Development of Long-term Biomonitoring in Elk Island National Park Using Benthic Macroinvertebrates

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DEVELOPMENT OF LONG-TERM BIOMONITORING IN ELK ISLAND NATIONAL PARK USING BENTHIC MACROINVERTEBRATES

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ABSTRACT

The biomonitoring of aquatic systems has increased substantially in recent years. Evidence of this can be seen in the number of specialised technical publications and symposia, in particular using benthic macroinvertebrates for aquatic biomonitoring. Benthic macroinvertebrates provide an integrated assessment of the temporal and spatial changes in the physical and chemical environment of a water body. These organisms also provide an early warning of detrimental changes in the structure and function of aquatic ecosystems because of their importance as a food source in a typical food web (e.g., microorganisms to invertebrates to fish, humans and other vertebrates) and in the breakdown and transfer of organic material (e.g., decomposition and nutrient cycling processes).

Many governments, especially in North America and Europe, are developing or have established biomonitoring programs using benthic macroinvertebrates. In Alberta benthic macroinvertebrates have often been used by the Department of Environmental Protection to monitor the effects of contaminants in rivers. In lentic systems, however, macroinvertebrates have not been routinely used as a biomonitoring tool. Staff at the Elk Island National Park (EINP) requested assistance from the Alberta Environmental Centre to determine the potential use of benthic macroinvertebrates for long-term biomonitoring and to establish baseline data on the aquatic invertebrate fauna in the park.

After discussions with EINP staff it was decided (1) to review the published literature on long-term biomonitoring using benthic macroinvertebrates in lentic habitats, (2) to conduct an inventory of the benthic macroinvertebrates in the littoral habitats of two lakes that were affected differently by human activities in the past, and (3) to recommend long-term biomonitoring method(s) based on the results of the first two objectives that might be used in EINP.

The literature searches produced several hundred document titles that included some form of biomonitoring or assessment in aquatic systems. A number of trends were evident from the literature examined.

(1) Most biomonitoring methods were developed for use in lotic instead of lentic systems, and most methods used in lentic systems concerned large water bodies, such as the Great Lakes.

(2) Several new biomonitoring methods using benthic macroinvertebrates were developed recently, are under development, or are already in use but are being modified for use in different countries than where the method was originally developed.

(3) Few of the biomonitoring methods, with the exception of newer methods, made direct reference to their use in long-term studies (over a period of decades). In many cases biomonitoring has not been conducted using consistent methods from year to year and, therefore, the data cannot be used in determining long-term trends.

The review of the literature included a summary of the biomonitoring methods, in particular Canadian studies conducted in lentic systems, using (1) individual organisms and populations of a taxon, (2) communities, or assemblages, of taxa, and (3) the role of macroinvertebrates in the benthic processes (such as energy transfer, nutrient cycling, and community metabolism) of aquatic ecosystems.

The main finding of the macroinvertebrate study in EINP was that there were clear differences between the macroinvertebrate fauna in the littoral habitats of the lakes. The differences between the abundances of the common macroinvertebrate taxa suggest that the shallow-water habitats of Astotin Lake were more eutrophic (high nutrient levels and low dissolved oxygen concentrations) than Bailey Lake. (In Astotin lake it is possible that enrichment was caused by the wastewater from the sewage treatment plant.) Nutrients and dissolved oxygen levels were not measured in the lakes and, therefore, this result is not conclusive. The results, however, illustrate the usefulness of using more than one macroinvertebrate taxon with known environmental requirements for determining the likely causes of differences between the lakes.

The following recommendations on the use of benthic macroinvertebrates for long-term biomonitoring in EINP were made.

(1) Identify the stressors, including the sources of contaminants, that are likely to cause changes in the aquatic systems. Include a broad survey of suspected contaminants (pesticides, fertilisers, and air-borne pollutants) in different aquatic media (sediment and biota).

(2) Identify and determine the importance of the aquatic habitat types in the park, including the unique habitats like the "soap holes".

(3) Determine the long-term management goals of EINP and how they will affect the aquatic systems in the park. This data will help decide on the habitat types to include in the biomonitoring program.

(4) A community-level method is recommended for long-term monitoring in EINP. Organism, population, or ecosystem-level methods are not recommended for biomonitoring at this time because many of these methods require prior knowledge on contaminants and stress-response effects on the biota. If contaminant concentrations are found to be important in the biota and sediment (see recommendation 1), the use of organism and population-level methods, such as sentinel organisms, might be more appropriate.

1 INTRODUCTION

The biomonitoring of aquatic systems has increased substantially in recent years. Evidence of this can be seen in the number of specialised technical publications and symposia, in particular using benthic macroinvertebrates for aquatic biomonitoring (e.g., Hellawell 1986, Evans 1988, Metcalfe and Reynoldson 1989, Plafkin *et al.* 1989, Klemm *et al.* 1990, Mckenzie *et al.* 1992, Environment Canada 1993, Gibbons *et al.* 1993, Rosenberg and Resh 1993, Loeb and Spacie 1994, Bunn 1995, Norris and Norris 1995). Many governments, especially in North America and Europe, are developing or have established biomonitoring programs using benthic macroinvertebrates (Metcalfe 1989, Plafkin *et al.* 1989, Hughes *et al.* 1992, Whittier and Paulsen 1992, Cairns and Pratt 1993, Environment Canada 1993, Ghetti and Ravera 1994, Paulsen and Linthurst 1994, Norris and Norris 1995).

Benthic macroinvertebrates include the arthropods (e.g., insects, crustaceans and arachnids) and other common organisms (e.g., snails, clams, leeches and aquatic earthworms) that live, for at least part of their life cycle, in the bottom substratum of lotic (flowing) and lentic (standing) freshwater systems. Macroinvertebrate taxa are used as biomonitors because (1) they are common in most freshwater systems, (2) they include a large number of taxonomic groups that show a range of responses to changes in the water and sediment quality, (3) they live in close association with and often feed on the bottom substratum, (4) they are relatively sessile in contrast to plankton and fish, and (5) many species have aquatic life stages of one or more years (Hellawell 1986, White 1988, Metcalfe 1989, Cairns and Pratt 1993, Johnson *et al.* 1993). Therefore, benthic macroinvertebrates provide an integrated assessment of the temporal and spatial changes in the physical and chemical environment of a water body. These organisms also provide an early warning of detrimental changes in the structure and function of aquatic ecosystems because of their importance as a food source in a typical food web (e.g., microorganisms to invertebrates to fish, humans and other vertebrates) and in the breakdown and transfer of organic material (e.g., decomposition and nutrient cycling processes).

In Alberta benthic macroinvertebrates have often been used by the Department of Environmental Protection (and its precursor, Alberta Environment) to monitor the effects of contaminants in rivers. In lentic systems, however, macroinvertebrates have not been routinely used as a biomonitoring tool (P. Mitchell, personal communication, Alberta Environmental

Protection). The monitoring of lakes in Alberta generally includes the measurement of chemical parameters, such as nutrients, ions, and chlorophyll *a* (as a measure of phytoplankton biomass).

Staff at the Elk Island National Park (EINP) requested assistance from the Alberta Environmental Centre to determine the potential use of benthic macroinvertebrates for long-term biomonitoring (i.e., over decades) in the park. The management goals of EINP require that the biota and integrity of habitats in the park are maintained for future generations (Olson 1993). The EINP staff were also seeking to establish baseline data on the aquatic invertebrate fauna in the park. The baseline data could be used to determine the common invertebrate taxa and to help with choosing appropriate taxa and biomonitoring methods.

The surface waters in EINP are mostly lentic systems and intermittent streams at the headwaters of five small drainages of the North Saskatchewan River basin. These headwaters are located in the Buffalo, or Cooking, Lake moraine (Griffiths 1992). The lentic habitats are mostly made up of shallow lakes and wetland types including bogs, fens, and marshes (Olson 1993). In addition to surface runoff, the effect of groundwater on the water chemistry of the aquatic systems is likely to be important because of the number of headwater streams in the park.

Potential disturbances and contaminants in the aquatic systems of the park include the deposition of air-borne contaminants (e.g., metals and trace organics) from Fort Saskatchewan and Edmonton, wastewater from the sewage treatment plant at Astotin Lake, and runoff from Highway 16 (e.g., salt and spills), the golf course, and agricultural land surrounding the park (e.g., fertilisers and pesticides) (McDonald *et al.* 1988). Changes in climate characteristics, such as the predicted increases in water temperatures and levels of ultraviolet radiation caused by reduced ozone levels, are also likely to have an effect on aquatic biota. The long-term effects, if any, of these factors on the aquatic systems in the park are unknown.

After discussions with EINP staff it was decided (1) to review the published literature on long-term biomonitoring using benthic macroinvertebrates in lentic habitats, (2) to conduct an inventory of the composition and abundance of the common benthic macroinvertebrates in the littoral habitats close to the outlets of Astotin Lake, where there are sources of contaminants, and Bailey Lake which generally has not been affected by human activities, and (3) to recommend long-term biomonitoring method(s) based on the results of the first two objectives that might be used in EINP.

2 LITERATURE REVIEW

Literature searches were conducted in nine computer databases to find references on biomonitoring in freshwater habitats, in particular those using aquatic invertebrates as long-term monitors in lentic habitats. The databases and the periods searched were: Aquatic Sciences and Fisheries Abstracts 1978-1995, Water Resources Abstracts 1968-1994, CAB Abstracts 1972-1995, BIOSIS Previews 1969-1995, NTIS 1964-1995, TOXLINE 1965-1995, Academic Index 1976-1995, CRIS/USDA 1995, and Energy SciTec 1974-1995 (Marshall 1993). The results of the searches were reviewed to determine the pertinent references that were used as the basis of the literature review. Additional references from the specialised scientific literature on biomonitoring (references cited in section 1) were also used as sources of information.

The literature searches produced several hundred document titles that included some form of biomonitoring or assessment in aquatic systems. The abstracts, authors, and publication details of each document were screened to determine pertinent documents that were obtained. Several documents, however, were not obtained, including articles not easily obtained through the inter-library loan service (Marshall 1993). These documents generally were not published in peer-reviewed scientific publications, such as journals, books, and proceedings.

A number of trends were evident from the literature examined.

(1) Most biomonitoring methods were developed for use in lotic instead of lentic systems, and most methods used in lentic systems concerned large water bodies, such as the Great Lakes.

(2) Several new biomonitoring methods using benthic macroinvertebrates were developed recently, are under development, or are already in use but are being modified for use in different countries than where the method was originally developed.

(3) Few of the biomonitoring methods, with the exception of newer methods, made direct reference to their use in long-term studies (over a period of decades). In many cases biomonitoring has not been conducted using consistent methods from year to year and, therefore, the data cannot be used in determining long-term trends.

Methods of biomonitoring using benthic macroinvertebrates include the evaluation of the effect of "stressors" or disturbances on (1) individual organisms and populations of a taxon, (2) communities, or assemblages, of taxa, and (3) the role of macroinvertebrates in the benthic processes, such as energy transfer, nutrient cycling, and community metabolism, of aquatic ecosystems. Examples of "stressors" affecting aquatic biota are physical (e.g., temperature, water level, and sediment) and chemical (e.g., nutrient enrichment and contaminants) changes caused by anthropogenic and natural processes.

As noted already in section 1 there are several recent reviews of the literature on benthic macroinvertebrates as biomonitors. Instead of repeating the content of this published work, this review will focus on a brief summary of methods in each of the three main groups of biomonitoring (stated above), and in particular, in those that have been applied in Canadian lentic systems. The discussion of lentic habitats will focus on littoral habitats that are typical of the shallow lakes which are common in EINP. The littoral habitat is defined as the shallow water where submergent vegetation is usually abundant. Wetlands *per se*, including bogs, fens, and marshes (Zoltai 1987), are not covered in the review.

2.1 Organism-level and Population-level Methods

Effects of stressors on individuals and populations of a taxon include biochemical, physiological, and behavioural responses, the occurrence of morphological deformities and contaminants in body tissues, and changes to the life history characteristics of organisms.

Biochemical and physiological methods include a wide range of methods, such as changes in enzyme activity, ion regulation, energy metabolism, and respiration rate caused by stressors in the environment (e.g., Giesy *et al.* 1988, Johnson *et al.* 1993). In the recent review of these and related methods using benthic macroinvertebrates, Johnson *et al.* (1993) concluded that many of these show potential for the biomonitoring of freshwater habitats, but there was a lack of data on the natural variation of the measurements and the effect of different types and levels of stressors on a range of taxa. In addition, the methods were often developed in the laboratory and they were rarely related to the results of field studies. Giesy *et al.* (1988) stated that most biochemical methods have not been developed for use in monitoring programs and will not replace population-level, community-level, or ecosystem-level monitoring. In a summary of a workshop on the biological monitoring of water quality in Australia, Bunn (1995) also stated that little attempt has been made to link biomarkers at the organism-level to real impacts in aquatic systems. The lack of data on these methods, in particular the biochemical methods, is understandable because they are relatively recent developments.

Behavioural changes will provide an early warning of sublethal effects on organisms. In toxicity tests, for example, the abnormal swimming behaviour of *Daphnia magna*, a common aquatic invertebrate in lentic habitats, can be used to determine the sublethal effects (EC_{50} values) of contaminants. Although behavioural studies can be conducted more easily in the laboratory than in the field (e.g., Rand 1985), field studies have successfully used behavioural changes, such as avoidance and feeding anomalies, to determine sublethal effects of stressors. Examples of field studies include the activity and molting of *Hexagenia* nymphs (Ephemeroptera)(Henry *et al.* 1986), valve movement of fingernail clams (Mollusca)(Doherty *et al.* 1987), and the feeding behaviour of Chironomidae larvae (Diptera)(Heinis *et al.* 1990).

Another biomonitoring method that can be used in both organism and population studies is the study of morphological deformities. Deformities have been reported in a range of benthic macroinvertebrates including the aquatic stages of insect taxa - midge (Chironomidae), stoneflies (Plecoptera), and caddis flies (Trichoptera) - and aquatic earthworms (Oligochaeta) in both lotic and lentic systems (Brinkhurst *et al.* 1968, Hamilton and Saether 1971, Hare and Carter 1976, Simpson 1980, Warwick 1980, Cushman 1984, Wiederholm 1984, Dickman *et al.* 1992). Most of the published reports of morphological deformities of invertebrates have been found in the mouthparts, antennae, and other structures of the head capsules of Chironomidae larvae in lentic habitats (see references cited in the reviews by Warwick 1988 and Johnson *et al.* 1993). In lentic systems Chironomidae larvae are often the most abundant benthic macroinvertebrates and include the greatest number of species of a macroinvertebrate taxon (Coffman and Ferrington 1984, Williams and Feltmate 1992, Thorp and Covich 1991).

In field studies the greatest proportion of organisms with morphological deformities have been located closest to sources of contaminants, that have included industrial and municipal wastewaters, and agricultural runoff (Warwick 1988, Johnson *et al.* 1993: Table 4.8). Differences between "contaminated" and "uncontaminated" sites, however, have not always been clear making it difficult to determine the cause of deformities. For example, the reported ranges in the proportion of deformities in benthic macroinvertebrate taxa varied from 0 to 100% at contaminated sites and 0 to 16% at relatively uncontaminated sites (Warwick 1988: Table 14.2, Johnson *et al.* 1993: Table 4.8). There are few clearly documented studies of the causes of the morphological deformities (Johnson *et al.* 1993). But in one laboratory study Kosalwat and Knight (1987) showed significantly greater incidences of deformities in a Chironomidae larvae with increasing concentrations of copper.

In Alberta there is only one published report of morphological deformities. Donald (1980) found a greater number of morphological deformities in adult stoneflies (Plecoptera) at sites downstream of domestic and industrial sewage outfalls when compared to cleaner sites on the Bow River. The Plecoptera larvae live in the Bow River and emerge from the river as adults. More recently, as part of studies conducted by the Northern River Basins Study, no morphological deformities in the head capsules of Chironomidae larvae were found in a preliminary examination of specimens from sites upstream and downstream of a pulp mill on the upper Athabasca River (R.L.&L. Environmental Services Ltd 1993).

Morphological deformities of benthic macroinvertebrates have also been used in paleolimnological studies (e.g., see references cited in Walker 1993 and Charles *et al.* 1994). Greater percentages of deformities in the head capsules of Chironomidae larvae have been found in sediments that were less than 50 years old compared to larvae in sediments of fifty to thousands of years old, when man-made chemicals were unknown or not widely used (Warwick 1980, Klink 1989, Walker 1993). Other paleolimnological methods based on the skeletal remains of a wide range of macroinvertebrates (including Insecta, Mollusca, Crustacea, Bryozoa, Oligochaeta, and Porifera) in aquatic sediments have been used to show changes in the aquatic environments caused by eutrophication, acidification, and climatic conditions (Walker *et al.* 1991, Williams and Feltmate 1992, Walker 1993, Charles *et al.* 1994).

Although, contaminants have been the suspected cause of morphological deformities in aquatic invertebrates, it is not known what effect a gradual change, such as increased water temperature and ultraviolet radiation associated with global warming, might have on the incidence of deformities. Warwick (1988), for example, suggested that changes in water temperature, such as, that caused by warm water effluents from cooling plants, might have caused morphological deformities in Chironomidae larvae in European and Canadian studies. In one Canadian study, Bothwell *et al.* (1994) found that near-ultraviolet radiation inhibited the colonization of substrata by Chironomidae larvae.

Benthic macroinvertebrates are often used to determine the concentrations and the fate and effects of contaminants in aquatic systems (Phillips 1980, Hellawell 1986, Nalepa and Landrum 1988, White 1988, Metcalfe and Reynoldson 1989, Hare 1992, Johnson *et al.* 1993, Phillips and Rainbow 1993). The concentrations of contaminants in the body tissues of macroinvertebrates are used to determine the levels of contaminants that are available to the resident biota. The concentration of contaminants can increase at higher trophic levels of the food chain. Major contaminants, such as organo-chlorine compounds (e.g., pesticides, polychlorinated biphenyls) and trace metals (e.g., copper, nickel, lead, and mercury) are generally persistent in the aquatic environment (Phillips 1980, Helawell 1986, White 1988).

Biota that concentrate contaminants from their environment are often referred to as sentinel or indicator organisms (Hellawell 1986, Johnson *et al.* 1993). Characteristics of sentinel organisms are given in the following list (Phillips 1980, Helawell 1986, Johnson *et al.* 1993).

(1) Individuals of the species show the same correlation between the contaminant in the organism and the environment at all locations and under all conditions.

(2) Individuals are capable of long-term reproduction even when exposed to maximum levels of the contaminant in the environment.

(3) The species is sedentary.

(4) The species is widespread in distribution, allowing comparisons between different areas.

(5) Sufficient tissue is available for analysis.

(6) The species is long-lived, enabling the sampling of several year-classes and the measurement of long-term effects.

(7) Individuals are easy to collect.

(8) Individuals are able to survive laboratory and field handling.

Although these characteristics are rarely found in the same aquatic organism, benthic macroinvertebrates are good candidates as sentinel organisms because they have many of these traits and they live in close association with sediments where contaminants often accumulate.

Specific examples of macroinvertebrates used to monitor the levels of contaminants in Canada include the use of aquatic insect larvae and adults (Ciborowski and Corkum 1988, Gobas *et al.* 1989, Hare 1992), bivalves and leeches (Green *et al.* 1989, Metcalfe and Hayton 1989), and other benthic macroinvertebrate taxa (Wagemann *et al.* 1978, Nalepa and Landrum 1988, White 1988).

Biomonitoring methods that use life history characteristics, such as survival, growth, and reproduction are restricted to studies on populations rather than individual organisms (Johnson *et al.* 1993). Like the organism-level methods, these methods can be used to determine sublethal effects on aquatic biota. Many chronic toxicity methods, for example, use life history traits to measure the sublethal effects of stressors (Petrocelli 1985, Burton *et al.* 1992, Buikema and Voshell 1993). When using life history characteristics for biomonitoring, it is essential to have background knowledge on the life history and biology of the species so that changes in the endpoints are attributed to the stressor and not to the natural variability within the population; in addition, food quality and habitat conditions can vary greatly within and among aquatic systems and these factors can have profound effects on life history traits (Johnson *et al.* 1993).

Other factors that affect the life history traits of macroinvertebrates include dissolved oxygen, dissolved solids, and temperature (Sweeney 1984, Wiederholm 1984, Johnson *et al.* 1993). Adult size and fecundity of aquatic insects, for example, might be used in the monitoring of the sublethal effects of temperature changes (Wiederholm 1984). In Alberta, Folsom and Clifford (1978) found flatworms to be twice as fecund in a thermally enriched area of Lake Wabamun compared to a natural lake. Other Canadian studies using life history traits include the effect of acidification and eutrophication on two common amphipods, *Hyalella azteca* and *Gammarus lacustris*, of lentic systems. In one study the reproductive success and survival of *H. azteca* were negatively affected by increased acidification and the effect was related to their size and development stage (France and Stokes 1987, Schindler 1990). In another study fertilization of an arctic lake caused increased phytoplankton production and survival of *G. lacustris* (Jorgenson *et al.* 1992).

2.2 Community-level Methods

The second major group of biomonitoring methods uses communities or assemblages of benthic macroinvertebrates to determine the ecological effects of environmental changes in aquatic systems (Helawell 1986, Metcalfe 1989, Johnson *et al.* 1993, Loeb and Spacie 1994). Community measures summarize and interpret data on the structure and function of aquatic ecosystems. The community methods include a large number of indices and ratios, multivariate statistical methods, and semi-quantitative methods.

The main types of indices include biotic, saprobic, diversity, and similarity indices (Washington 1984, Hellawell 1986, Metcalfe 1989, Johnson *et al.* 1993). Biotic indices are based on the resident species and their tolerances for specific environmental conditions, particularly organic pollution. Therefore, these indices are generally specialised for specific geographical areas (Washington 1984, Hellawell 1986, Hilsenhoff 1988, Johnson *et al.* 1993). Saprobic indices are based on the presence of indicator species that have been assigned scores based on their tolerance of decomposing organic material and dissolved oxygen conditions (Metcalfe 1989, Johnson *et al.* 1993).

Diversity indices combine the number of species (richness), uniformity of the distribution of each species (evenness), and the total number of organisms in the community into a single number to reflect changes in the community caused by environmental stress (Washington 1984, Hellawell 1986, Metcalfe 1989). In general, lower biotic and diversity indices indicate stressed systems (Johnson *et al.* 1993, Norris and Georges 1993). Species richness, however, is sometimes more biologically meaningful than the other indices, in particular, the diversity indices, because the index values are difficult to interpret (Washington 1984, Metcalfe 1989, Johnson *et al.* 1993). In several studies species richness has been shown to be a more reliable indicator of water quality than diversity indices (e.g., Lenat 1983). Similarity indices involve the ordering of samples relative to overall similarities and then examining the major gradients for correlations with environmental factors; similarity indices can only be used when there is a reference (or control) site for comparison with "altered" sites (Washington 1984, Hellawell 1986).

Multivariate statistical methods, another major group of community-level methods, can be used to summarise the effect of physical and chemical factors on biota and, in some cases, predict the benthic macroinvertebrate community based on physical and chemical parameters of the habitat (e.g., Wright *et al.* 1984, Ormerod and Edwards 1987, Moss *et al.* 1987, Jackson 1993, Norris and Georges 1993 and Johnson *et al.* 1993). These analyses include a number of classification and ordination statistical procedures, such as cluster analysis, principal components analysis, detrended correspondence analysis, and canonical correspondence analysis (e.g., Pielou 1984). The use of multivariate procedures with benthic macroinvertebrate data is reviewed by Johnson *et al.* (1993) and Norris and Georges (1993).

Semi-quantitative biomonitoring methods of benthic macroinvertebrate communities have been developed recently and used extensively, particularly, in the U.S.A., (e.g., Ohio Environmental Protection Agency 1989, Plafkin *et al.* 1989, Klemm *et al.* 1990, Resh and Jackson 1993, Gurtz 1994, Lenat and Barbour 1994). These methods use indices, or "metrics", to describe the macroinvertebrate community, often at a higher taxonomic level than is usual for quantitative studies. Statistical procedures are rarely used in these methods; examples are the Invertebrate Community Index and Rapid Bioassessment Protocols (Ohio EPA 1989, Plafkin *et al.* 1989, Klemm *et al.* 1990, Resh and Jackson 1993). Two other semi-quantitative methods that include benthic macroinvertebrate communities as a component of national monitoring programs in the U.S.A. are the Environmental Monitoring and Assessment Program (EMAP) and the National Water-Quality Assessment Program (NAWQA) developed by the U.S. EPA and U.S. Geological Service, respectively (Whittier and Paulsen 1992, Gurtz 1994, Paulsen and Linthurst 1994). These programs were set up as long-term monitoring programs of aquatic systems.

In Canada, unlike the U.S.A., no national biomonitoring programs have been established for the long-term monitoring of trends in aquatic systems. The Environmental Effects Monitoring (EEM) program, however, was developed by Environment Canada to provide general guidance and recommendations on methods, including the sampling of benthic macroinvertebrate communities, to assess the adequacy of the effluent regulations under the federal Fisheries Act (Environment Canada 1993). In another federal biomonitoring program the Department of Fisheries and Oceans is using the benthic macroinvertebrate community of the littoral area of lakes to assess the long-term effects of acidic deposition in central and eastern Canada (Davies 1991). Davies found the number of species in a chironomid community declined by almost one-half after the artificial acidification of an Ontario lake. In the Great Lakes, benthic macroinvertebrate communities were successfully used to monitor long-term changes in water quality but only when consistent field techniques and sampling times were used (Barton 1989).

2.3 Ecosystem-level Methods

Benthic macroinvertebrates play an important role in the functioning of aquatic systems, in particular in the processes of decomposition and production, such as nutrient cycling and energy transfer between trophic levels, and community metabolism. For example, the principal site of decomposition in aquatic systems is in the sediment where dead organisms and faeces settle. Changes in the structure of the benthic community can be used to make inferences about ecosystem-level processes (Reice and Wohlenberg 1993).

Ecosystem-level manipulations in lotic and lentic systems have demonstrated that benthic macroinvertebrates are especially sensitive when compared to other freshwater biota, and they can be used for monitoring ecosystem changes (e.g., Schindler 1990, Reice and Wohlenberg 1993, Bothwell *et al.* 1994). For example, in the Experimental Lakes Area, in central Canada, artificial acidification of Lake 223 eliminated acid-sensitive organisms, especially large benthic crustaceans such as *Hyalella azteca* and these organisms were an important food source of lake trout (Schindler *et al.* 1985, Schindler 1990). After acidification populations of lake trout, a top predator in the food web, remained at normal levels, although the trout showed increased starvation and mortality because of the elimination of it's main food source (Schindler *et al.* 1985). Major changes caused by acidification, therefore, were not evident in the size of the lake trout population, but *H. azteca*, that occupied key ecological niches, provided an early warning of a detrimental change within an ecosystem (Schindler 1990). In another whole-lake manipulation study in Wisconsin acidification affected the rates of decomposition of different species of leaf that would result in changes to ecosystem processes (Perry and Troelstrup 1988).

Another Canadian example of ecosystem-level manipulations and biomonitoring in lentic systems is the Marsh Ecology Research Program (MERP) at the Delta Waterfowl and Wetlands Research Station in Manitoba (Murkin 1989a). This multidisciplinary research program was developed to conduct long-term research on the effect of water level manipulations on northern prairie marshes. Although this program involves research on marshes, many of the results are applicable to the littoral areas of lakes. The research includes the extensive use of benthic macroinvertebrates to examine the effects of water level changes and processes of decomposition and interactions with terrestrial food webs (Murkin *et al.* 1983, Murkin 1989b, Campeau *et al.* 1994).

3 BENTHIC MACROINVERTEBRATE FAUNA IN ELK ISLAND NATIONAL PARK

3.1 Choice and Description of Study Sites

Drainage basins in the park are made up of the headwaters of five named creeks (Norris, Astotin, Oster, Ross, and Lamont; listed in decreasing order of the area of each stream basin within the park boundary) that are part of the North Saskatchewan River basin (Figure 1). Most of the streams in EINP are affected by beaver dams, and flows are intermittent throughout the year. Two lakes, Astotin and the southern lake of Bailey Lakes (hereafter referred to as Bailey Lake), that were affected differently by human activities in the past were chosen for the surveys of the benthic macroinvertebrate fauna (Figure 1). Both lakes are shallow with intermittent streams draining in and out of each lake. Similar habitat types close to the outlets of each lake were chosen for the zoobenthos surveys because the composition and abundance of the fauna in these areas would probably be affected by the overall water quality and cumulative effects in each lake (Figures 2 and 3).

Astotin Lake, at the headwaters of Astotin Creek, is one of the largest lakes in the park (Figure 1). Runoff from the golf course and wastewater from the sewage treatment plant might have affected the water quality and biota in Astotin lake. Recreational activities, including the use of unmotorized boats and a beach, also occur on Astotin Lake. In contrast, Bailey Lake, at the headwaters of Norris Creek, is a smaller lake located in the southern part of the park, south of Highway 16 (Figure 1). Bailey Lake on the whole has not been affected by human activities in the history of the park.

Close to the outlet of each lake there were extensive littoral zones containing abundant aquatic vegetation, including submergent (mostly *Myriophyllum exalbescens* Fern. and other less common taxa, such as *Elodea canadensis* Michx., *Ranunculus circinatus* Sibth, *Ceratophyllum demersum* L.), floating (*Polygonum natans* (Eaton), *Lemna trisulca* L.), and emergent (*Sparganium eurycarpum* Engelm., *Typha latifolia* L., *Sagittaria cuneata* Sheld.) plants (Burland 1994). In Astotin Lake there were also extensive beds of the macrophytic alga, *Chara*, in an

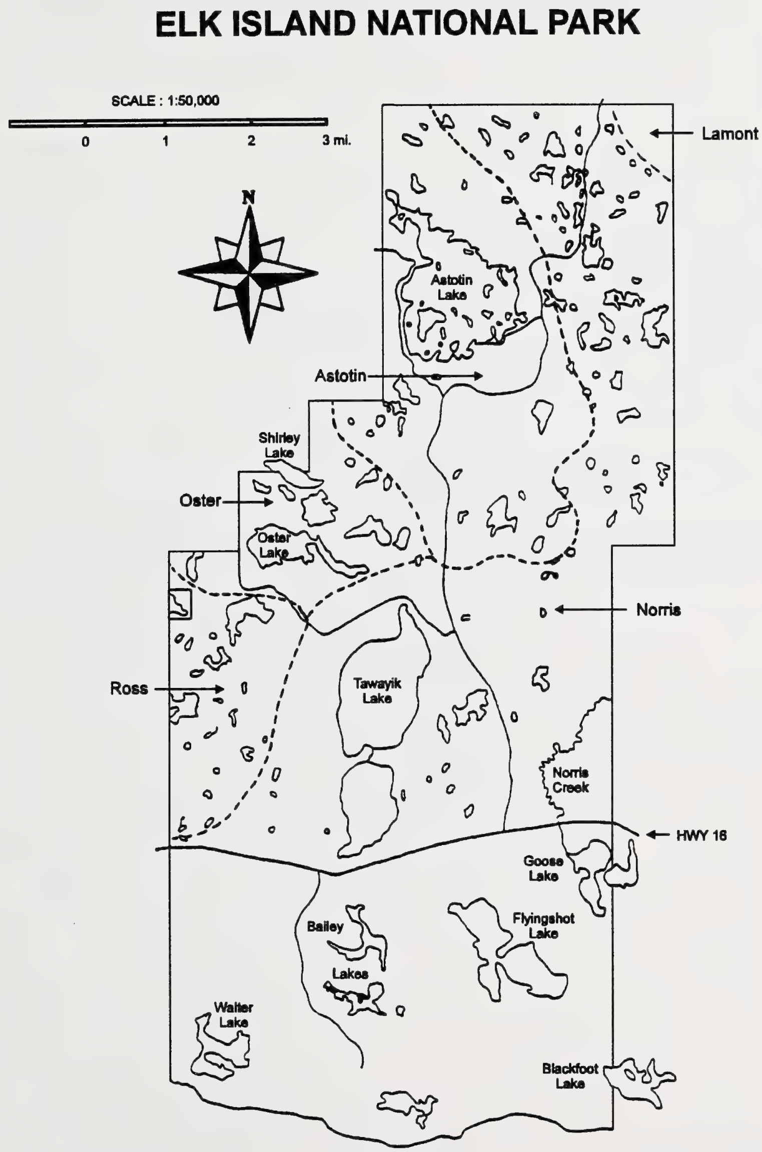


Figure 1. Major aquatic systems of Elk Island National Park showing the five drainage basins (Norris, Astotin, Oster, Ross, and Lamont).

Figure 2. Astotin Lake showing the location of the littoral habitat types.

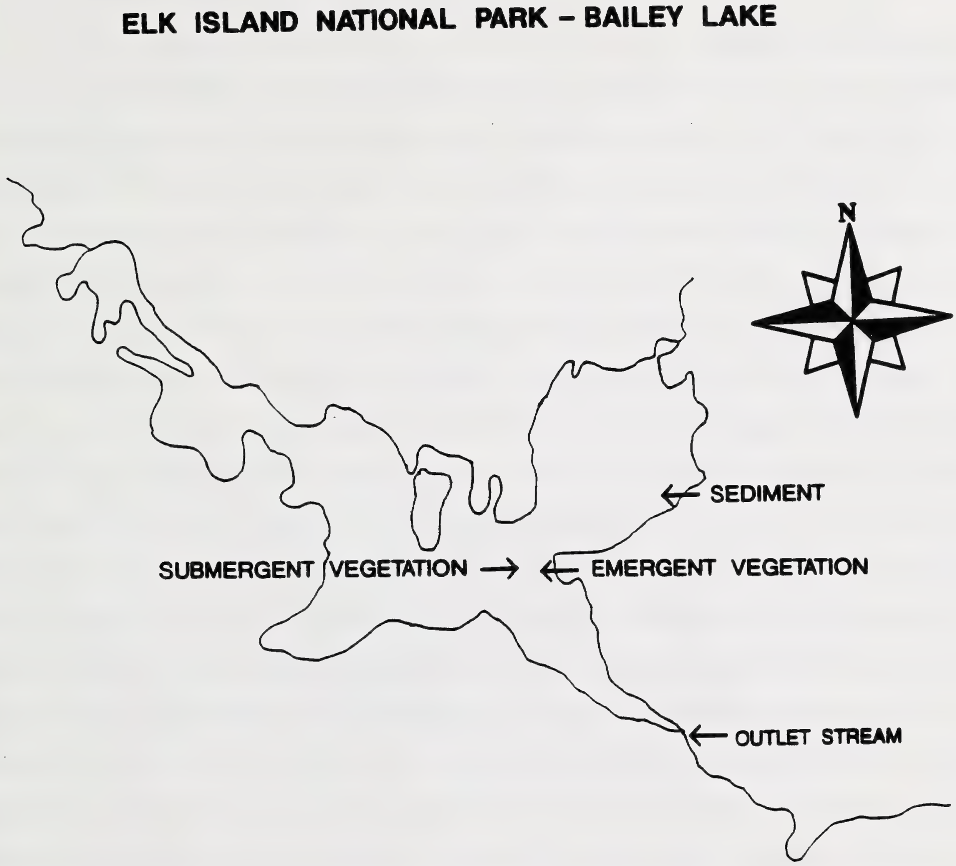


Figure 3. Bailey Lake showing the location of the littoral habitat types.

open water area close to the outlet (Figure 2).

3.2 Methods and Materials

Preliminary surveys of the shallow-water (<2 m) habitat types close to the outlets of each lake were conducted by canoe and on foot from the shoreline to choose similar or unique aquatic habitat types that were predominant in each lake. Three main types of habitat, (1) submergent and (2) emergent vegetation and (3) fine sediment with no vegetation, were found in each lake; a large area of *Chara* was the fourth habitat type that was only found in Astotin Lake (Figures 2 and 3). The submergent vegetation sites were located from about 4 to 7 m from the shoreline in water ranging from 27 to 92 cm deep. The emergent vegetation sites were located along the shoreline and amongst the reeds in water from 15 to 25 cm deep. The sediment sites were from about 14 to 35 m from shore in water from 85 to 130 cm deep. The *Chara* sites were about 40 to 50 m from the shore and water depths ranged from 127 to 132 cm. These four habitat types were typical of the shallow-water areas of the lakes in EINP.

All of the sites were sampled from a canoe to reduce disturbance to the habitat; the samples were taken by the same operator while the second person maintained the position of the boat. For each of the submergent and emergent vegetation and *Chara* habitat types, five samples were taken with a D-shaped dip net (dimensions of net opening = 30x33 cm, length of net = 50 cm, pore size of net = 0.5 mm; Merritt *et al.* 1984) using a semi-quantitative method. At each sample site a representative area of the habitat type was chosen before positioning the canoe alongside the site. The dip net was submerged in the water amongst the vegetation and immediately moved in three sweeping strokes from side to side within an approximate area of 1x1 m. The vegetation and *Chara* samples often contained quantities of the loosely consolidatedfine sediment from the base of the plants that was disturbed by the movement of the net. Most of the water was allowed to drain through the net before the vegetation samples were preserved in 95% ethanol. For the sediment habitats of each lake five samples were taken using the quantitative Ekman grab (sampling area = 15x15 cm, sampling depth = 15 cm; Merritt *et al.* 1984) and preserved in 10% formalin. The preservative in all samples was replaced within three weeks with 85% ethanol. For each of the four habitat types sampled, only three of the five samples were analysed in the laboratory. The samples were taken in both lakes from 15 to 25

August 1994. At this time of the year most macroinvertebrate taxa with aquatic stages are abundant in littoral habitats.

In the laboratory the samples were analysed using the same general procedure with modifications depending on the size and type of material in the sample. Each of the submergent and emergent vegetation and *Chara* samples were divided into three size fractions (coarse: >11.8 mm, medium: 11.8-0.21 mm, and fine: <0.21 mm) by washing the sample through sieves. Because of the large size and the time required to process the sample fractions, in particular the fine fractions, subsamples were taken in many of the coarse and medium size fractions. The subsamples were randomly taken from the sample fractions after they were thoroughly mixed in a container of water. For the coarse fractions most of the macroinvertebrates and organic material were decanted from the sample mixture in a 5-gallon pail and three 200 ml subsamples were taken from the remaining sample material. The combined size of the three subsamples in the coarse fractions ranged from 12 to 75% of the sample volume. For the medium fraction three 50 ml subsamples, ranging from 8 to 30% of the sample volume, were taken from the material in the 0.21 mm sieve. All of the fine fractions in the vegetation and *Chara* samples were subsampled based on the method of Wrona *et al.* (1982). Three 12 ml subsamples, ranging from 1 to 36% of the total sample volume, were taken from the sample in the Imhoff cone. For the Ekman grab samples of the sediment, the size of the particles in the samples was generally <0.210 mm. Subsamples, ranging from 4 to 6% of the sample volume, were taken in all of the sediment samples (Wrona *et al.* 1982). The macroinvertebrates were extracted from the samples at ≥ 60x magnification using the dissecting microscope and stored in vials of 85% ethanol.

The macroinvertebrates were identified using the *Aquatic Invertebrates of Alberta* (Clifford 1991) as the main taxonomic reference and other pertinent references when necessary (e.g., Merritt and Cummins 1984, Pennak 1989, Thorp and Covich 1991). Many of the macroinvertebrate taxa were not identified beyond elementary taxonomic levels because of the additional resources required, including the assistance of taxonomic specialists.

3.3 Results

Combining the data for both lakes, three taxa, *Hyalella azteca* (Saussure)(Crustacea: Amphipoda), Chironomidae (Diptera), and Oligochaeta (Annelida), made up 69%, 47%, 73%, and

85% of the total number of organisms in the submergent and emergent vegetation, *Chara*, and sediment samples, respectively (Tables 1, 2, 3, and 4). These three taxa and another amphipod, *Gammarus lacustris* Sars, showed clear differences in their relative abundances between the lakes (Figures 4, 5, and 6). For each of the vegetation and sediment habitat types, *H. azteca* and *G. lacustris* were more abundant in Bailey Lake than in Astotin Lake, whereas Chironomidae and Oligochaeta were less abundant in Bailey Lake relative to Astotin Lake (Figures 4, 5, and 6).

Other less common taxa each accounted for ≤12% of the total number of organisms in each habitat type (Tables 1, 2, 3, and 4). These taxa also showed differences in their relative abundances between the lakes. For example, in the submergent and emergent vegetation *Promenetus umbillicatellus* (Cockerell)(Mollusca: Gastropoda), Ceratopogoninae (Diptera: Ceratopogonidae), and *Caenis* (Ephemeroptera: Caenidae) were more abundant in Astotin Lake than in Bailey Lake, whereas, Copepoda (Crustacea) and *Physa* (Mollusca: Gastropoda) were more abundant in Bailey Lake than in Astotin Lake (Figure 4; Tables 1 and 2). Trends in the abundances of the remaining taxa were not as clear as those described above. The abundances of Ostracoda, Porifera, Hydrozoa, and semi-aquatic Collembola were not counted because they were often abundant and difficult to identify to lower taxonomic groups.

Comparing the abundance of the fauna in the lakes the total number of organisms were similar in the submergent and emergent vegetation but there was a two fold increase of the total number of organisms in the sediment of Bailey Lake compared to Astotin Lake (Figures 4, 5, and 6). The total number of organisms was approximately an order of magnitude less in the sediment than in the other habitat types of both lakes (Figures 4, 5, and 6).

3.4 Discussion

The main finding of the study was that there were clear differences between the macroinvertebrate fauna in the littoral habitats of the lakes. The differences between the abundances of the common macroinvertebrate taxa suggest that the shallow-water habitats of Astotin Lake were more eutrophic (having high nutrient levels and low dissolved oxygen concentrations) than Bailey Lake. Chironomidae and Oligochaeta, often found at large densities in nutrient enriched habitats and low dissolved oxygen concentrations (Pennak 1989, Brinkhurst and Gelder 1991, Hilsenhoff 1991), were at much greater abundances in Astotin Lake than in

Table 1. Macroinvertebrate taxa in the submergent vegetation samples from Astotin Lake and Bailey Lake. Larvae (L) and adults (A) are shown separately for the Coleoptera and Hemiptera.

TAXONOMIC GROUP	ASTOTIN LAKE				BAILEY LAKE				TOTAL NO. OF ORGANISMS	PERCENT COMPOSITION
	1	2	3	MEAN	1	2	3	MEAN		
INSECTA										
DIPTERA										
CHIRONOMIDAE	214	847	241	434	476	133	158	256	2069	12
CERATOPOGONIDAE										
CERATOPOGONINAE	68	230	102	133	4	20	3	9	427	3
COLEOPTERA										
HALIPLIDAE										
Haliplus(A)	2	0	0	1	0	0	0	0	2	0
Haliplus(L)	8	30	8	15	0	0	0	0	46	0
DYTISCIDAE										
HYDROPORINAE(A)	0	0	0	0	0	13	0	4	13	0
CURCULIONIDAE(A)	33	0	1	11	0	0	0	0	34	0
HEMIPTERA										
CORIXIDAE(A)	15	30	44	30	2	127	2	44	220	1
NOTONECTIDAE(A)	0	4	2	2	1	0	1	1	8	0
ODONATA										
ZYGOPTERA	28	25	5	19	85	10	32	42	185	1
ANISOPTERA	0	0	0	0	0	0	1	0	1	0
TRICHOPTERA										
LEPTOCERIDAE	2	0	0	1	0	0	0	0	2	0
PHYRGANEIDAE	2	51	21	25	2	0	2	1	78	0
EPHEMEROPTERA										
CAENIDAE										
Caenis	106	87	106	100	0	0	2	1	301	2
CRUSTACEA										
AMPHIPODA										
Hyalella azteca	114	214	500	276	515	2833	346	1231	4522	27
Gammarus lacustris	2	9	22	11	217	107	26	117	383	2
CLADOCERA	51	0	42	31	0	0	0	0	93	1
COPEPODA										
CYCLOPOIDA	40	0	10	17	19	133	0	51	202	1
HARPACTICOIDA	550	0	0	183	11	233	550	265	1344	8
CALANOIDA	0	0	0	0	4	0	0	1	4	0
HYDRACARINA	13	21	7	14	15	0	0	5	57	0
MOLLUSCA										
GASTROPODA										
PLANORBIDAE										
Promenetus umbilicatellus	197	300	65	187	2	10	0	4	574	3
Armiger crista	0	4	0	1	0	0	0	0	4	0
PHYSIDAE										
Physa	44	121	55	73	91	583	52	242	946	6
PELECYPODA										
SPHAERIDAE	0	26	21	16	0	33	0	11	80	0
ANNELIDA										
OLIGOCHAETA	2086	1778	1056	1640	48	53	124	75	5145	30
HIRUDINEA	9	13	51	24	13	87	15	38	188	1
TOTAL NO. OF ORGANISMS	3584	3790	2359	3244	1505	4376	1314	2398	16928	100

Table 2. Macroinvertebrate taxa in the emergent vegetation samples from Astotin Lake and Bailey Lake. Larvae (L) and adults (A) are shown separately for the Coleoptera and Hemiptera.

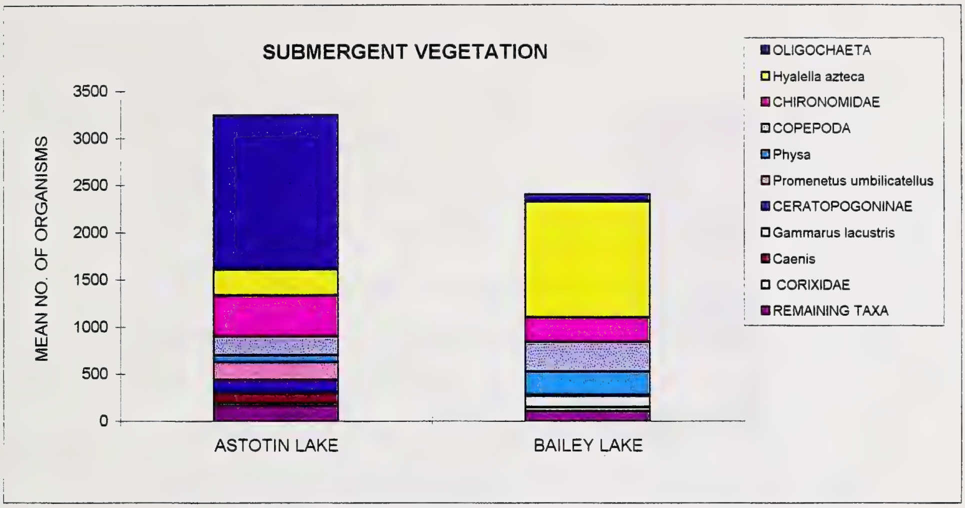
TAXONOMIC GROUP	ASTOTIN LAKE				BAILEY LAKE				TOTAL NO. OF ORGANISMS	PERCENT COMPOSITION
	1	2	3	MEAN	1	2	3	MEAN		
INSECTA										
DIPTERA										
CHIRONOMIDAE	803	300	500	534	0	57	69	42	1729	12
SYRPHIDAE	0	1	8	3	0	0	0	0	9	0
CERATOPOGONIDAE										
CERATOPOGONINAE	200	73	47	107	0	20	7	9	347	2
EPHYDRIDAE	0	0	0	0	0	0	3	1	3	0
COLEOPTERA										
HALIPLIDAE										
Haliplus (L)	30	13	20	21	13	0	0	4	76	1
Peltodytes(A)	5	0	0	2	0	0	0	0	5	0
Peltodytes(L)	0	0	7	2	0	0	0	0	7	0
DYTISCIDAE										
HYDROPORINAE(A)	15	0	0	5	13	7	0	6	35	0
HEMIPTERA										
CORIXIDAE(A)	124	128	109	120	13	0	21	11	394	3
GERRIDAE(A)	0	0	7	2	0	0	0	0	7	0
NOTONECTIDAE(A)	0	7	0	2	0	0	13	4	20	0
TRICHOPTERA										
PHYRGANEIDAE	0	7	0	2	0	0	3	1	10	0
EPHEMEROPTERA										
CAENIDAE										
Caenis	0	7	7	4	0	0	0	0	13	0
CRUSTACEA										
AMPHIPODA										
Hyalella azteca	524	173	133	277	850	1697	1161	1236	4538	32
Gammarus lacustris	6	1	0	2	75	73	168	105	323	2
CLADOCERA	556	0	63	207	0	200	381	194	1200	9
COPEPODA										
CYCLOPOIDA	950	250	150	450	100	1487	347	645	3284	23
HARPACTICOIDA	0	0	0	0	0	0	8	3	8	0
HYDRACARINA	160	57	0	72	0	0	6	2	222	2
MOLLUSCA										
GASTROPODA										
PLANORBIDAE										
Promenetus umbilicatellus	438	214	137	263	0	7	6	4	802	6
Promenetus exacuous	0	7	1	3	75	60	55	63	198	1
Helisoma	0	0	0	0	0	0	3	1	3	0
PHYSIDAE										
Physa	26	14	7	16	0	7	68	25	122	1
LYMNAEIDAE	0	1	7	3	13	0	10	8	31	0
VALVATIDAE	0	1	0	0	0	0	0	0	1	0
ANNELIDA										
OLIGOCHAETA	370	50	13	144	0	0	27	9	460	3
HIRUDINEA	77	14	30	40	0	20	7	9	147	1
TOTAL NO. OF ORGANISMS	4284	1318	1244	2282	1150	3634	2363	2382	13993	100

Table 3. Macroinvertebrate taxa in the *Chara* samples from Astotin Lake. Larvae (L) and adults (A) are shown separately for the Coleoptera and Hemiptera.

TAXONOMIC GROUP	1	2	3	MEAN	TOTAL NO. OF ORGANISMS	PERCENT COMPOSITION
INSECTA						
DIPTERA						
CHIRONOMIDAE	555	1620	492	889	2667	58
COLEOPTERA						
HALIPLIDAE						
Haliplus(A)	0	2	0	1	2	0
Haliplus(L)	53	77	25	52	155	3
CURCULIONIDAE(A)	1	0	0	0	1	0
Endalus(L)	0	1	0	0	1	0
HEMIPTERA						
CORIXIDAE(A)	72	42	103	72	216	5
NOTONECTIDAE(A)	0	1	0	0	1	0
TRICHOPTERA						
PHYRGANEIDAE	0	8	5	4	13	0
CRUSTACEA						
AMPHIPODA						
Hyalella azteca	0	1	1	1	2	0
Gammarus lacustris	0	1	1	1	2	0
HYDRACARINA	62	94	59	72	215	5
MOLLUSCA						
GASTROPODA						
PLANORBIDAE						
Promenetus umbilicatellus	183	264	95	180	541	12
PHYSIDAE						
Physa	17	16	16	16	49	1
LYMNAEIDAE						
Stagnicola	0	0	1	0	1	0
VALVATIDAE	0	2	0	1	2	0
PELECYPODA						
SPHAERIDAE	0	25	8	11	33	1
ANNELIDA						
OLIGOCHAETA	171	383	123	226	677	15
HIRUDINEA	0	1	1	1	2	0
TOTAL NO. OF ORGANISMS	1112	2537	929	1526	4579	100

Table 4. Macroinvertebrate taxa in the emergent vegetation samples from Astotin Lake and Bailey Lake. Larvae (L) and adults (A) are shown separately for the Coleoptera and Hemiptera.

TAXONOMIC GROUP	ASTOTIN LAKE				BAILEY LAKE				TOTAL NO. OF ORGANISMS	PERCENT COMPOSITION
	1	2	3	MEAN	1	2	3	MEAN		
INSECTA										
DIPTERA										
CHIRONOMIDAE	0	33	300	111	17	75	0	31	425	25
CRUSTACEA										
AMPHIPODA										
Hyalella azteca	0	0	0	0	83	625	140	283	848	50
Gammarus lacustris	20	0	0	7	67	100	0	56	187	11
HYDRACARINA	0	0	0	0	0	25	0	8	25	1
MOLLUSCA										
GASTROPODA										
PLANORBIDAE										
Armiger crista	0	0	0	0	0	0	20	7	20	1
PELECYPODA										
SPHAERIDAE	0	17	0	6	0	0	0	0	17	1
ANNELIDA										
OLIGOCHAETA	20	150	0	57	0	0	0	0	170	10
HIRUDINEA	0	0	0	0	17	0	0	6	17	1
TOTAL NO. OF ORGANISMS	40	200	300	180	183	825	160	384	1692	100



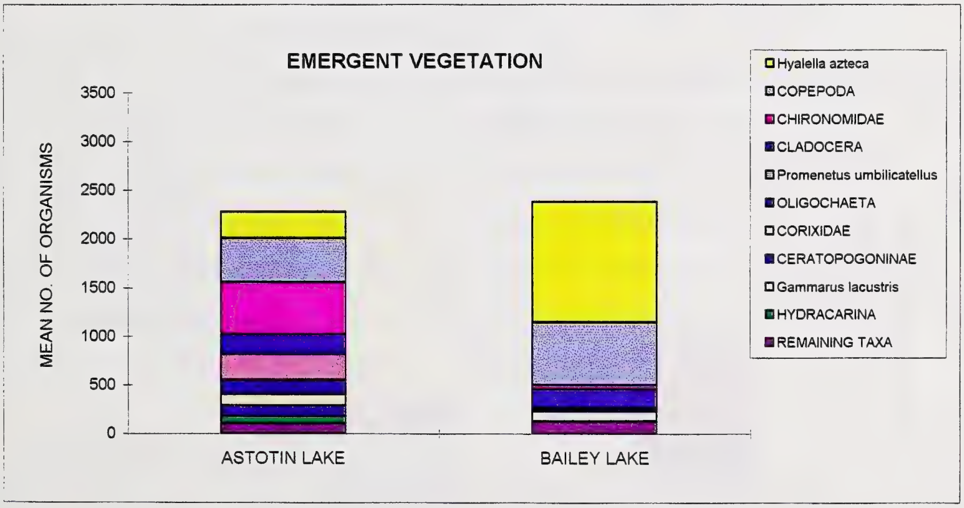


Figure 4. Mean number of organisms (n=3) for the macroinvertebrate taxa in the submergent and emergent vegetation samples from Astotin Lake and Bailey Lake.

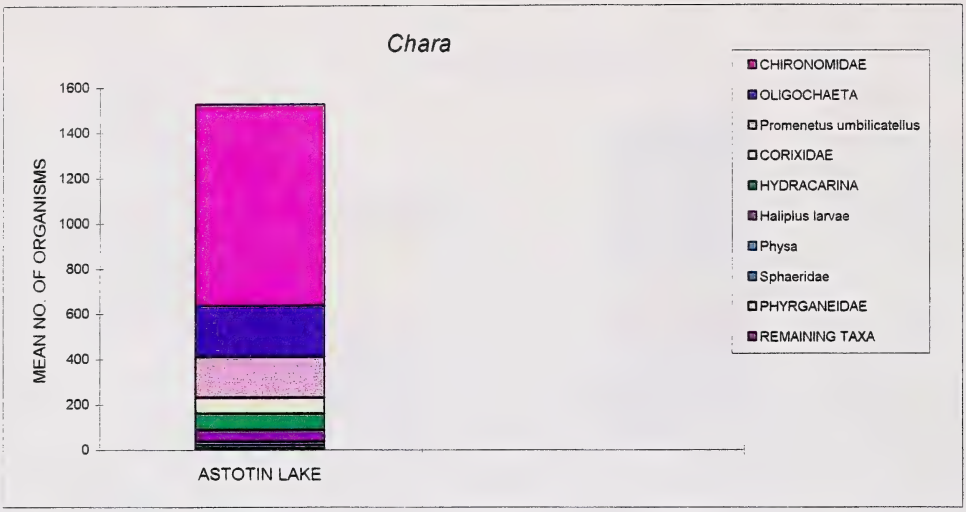


Figure 5. Mean number of organisms (n=3) for the macroinvertebrate taxa in the *Chara* samples from Astotin Lake.

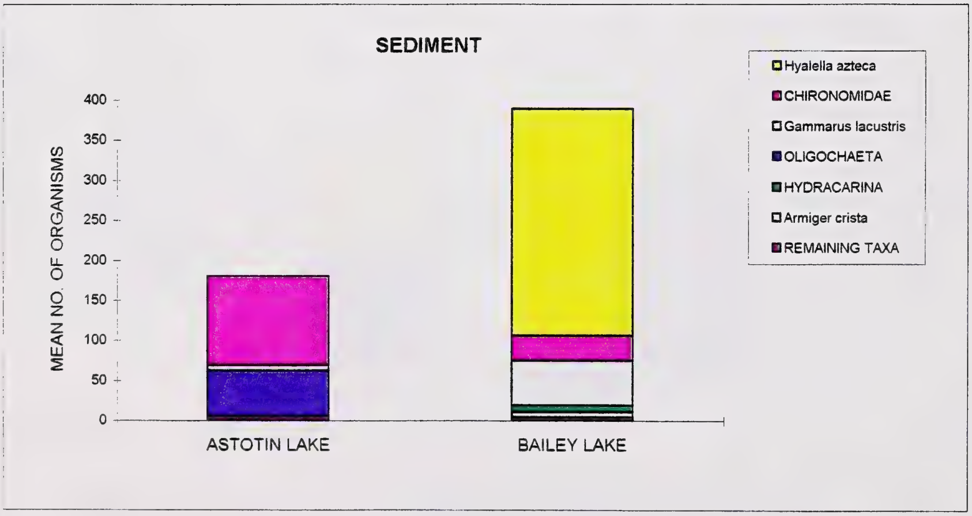


Figure 6. Mean number of organisms (n=3) for the macroinvertebrate taxa in the sediment samples from Astotin Lake and Bailey Lake.

Bailey Lake. Both the Chironomidae and Oligochaeta, however, also include species that are found in low nutrient levels and high dissolved oxygen concentrations (Coffman and Ferrington 1984, Brinkhurst and Gelder 1991, Hilsenhoff 1991). More conclusive results might be obtained by identifying the organisms to lower taxonomic levels, ideally the species-level.

The relative abundances of three other macroinvertebrate taxa in the lakes might also indicate eutrophic conditions in Astotin Lake. The clear differences in the abundances of the two species of Amphipoda, particularly *H. azteca* that was more common than *G. lacustris*, might be the result of differences in the concentrations of dissolved oxygen in the lakes. Both of these amphipods require high dissolved oxygen concentrations and are common in unpolluted waters (Pennak 1989, Covich and Thorp 1991). There are, however, cases where large numbers of these species have been collected in "eutrophic lakes" (de March 1981a, 1981b). *Caenis* larvae were also more abundant in Astotin Lake than in Bailey Lake. Species of *Caenis* are known to be tolerant of polluted conditions, including low dissolved oxygen concentrations (Edmunds 1976, Hilsenhoff 1991). Differences between the abundances of the other macroinvertebrate taxa are difficult to interpret because the environmental requirements or preferences of these taxa are not as well known (e.g., Pennak 1989, Thorp and Covich 1991).

In this study more eutrophic conditions were inferred in Astotin Lake relative to Bailey Lake by the distribution and abundance of the macroinvertebrate fauna. (In Astotin lake it is possible that enrichment was caused by the wastewater from the sewage treatment plant.) Nutrients and dissolved oxygen levels were not measured in the lakes and, therefore, this result is not conclusive. The results, however, illustrate the usefulness of using more than one macroinvertebrate taxon with known environmental requirements for determining the likely causes of differences between the lakes.

4 RECOMMENDATIONS

(1) Identify the stressors, including the sources of contaminants, that are likely to cause changes in the aquatic systems of the park. This could be done initially by reviewing the data on the surface waters and groundwater already available or being revised (e.g., McDonald *et al.* 1988, Olson 1993). Gaps in this database should be filled in by obtaining pertinent information. For example, some trace

organic contaminants have been found in the surface waters of the park, but there are no data on contaminant concentrations in the sediments or biota where contaminants are likely to accumulate and possibly biomagnify in the food webs. Include a broad survey of suspected contaminants (pesticides, fertilisers, and air-borne pollutants) in different aquatic media (sediment and biota).

(2) Identify and determine the importance of the aquatic habitat types in the park, including the unique habitats like the "soap holes" (Olson 1993). For example, determine if the headwater streams, that are common in the park, are fed mainly from groundwater or runoff from the surrounding land. This information can be used to determine the contribution of groundwater and runoff in the water budgets of the aquatic systems and to identify the potential sources and types of contaminants.

(3) Determine the long-term management goals of EINP and how they will affect the aquatic systems in the park. This data will help decide on the habitat types to include or exclude in the biomonitoring program. The biomonitoring program does not necessarily need to include all of the aquatic habitats in the park but choose representative habitats and those that are likely to be impacted.
Anticipated changes to the aquatic systems in the management plan can be used in choosing the most appropriate sites for biomonitoring. For example, changes to the biota (the re-introduction of native fish species to Astotin Lake), or changes to the physical and chemical environment (those caused by the management of beaver activities), will likely affect the results of future biomonitoring.

(4) A community-level method is recommended for long-term monitoring in EINP. Examination of the macroinvertebrate community will provide an integrated effect of environmental changes on a range of biota with different environmental requirements and at different trophic levels (consumers and predators) in aquatic food webs. Organism, population, or ecosystem-level methods are not

recommended for biomonitoring at this time because many of these methods require prior knowledge on contaminants and stress-response effects on the biota. If contaminant concentrations are found to be important in the biota and sediment (see recommendation 1), the use of organism and population-level methods, such as sentinel organisms, might be more appropriate. The semi-quantitative community methods are unlikely to be used successfully in EINP if the stressor(s) does not show a strong effect on the macroinvertebrate fauna. These methods are often used to show general changes in macroinvertebrate communities over major geographical regions.

Different multivariate statistical methods can be used to analyse the community data depending on the design of the biomonitoring program. The level of taxonomic identification is an important factor when considering the sensitivity of the community method (e.g., Chironomidae and Oligochaeta in the EINP study). Identification to the species level will provide the most sensitivity but it will also require more resources. The resources required for identification can be reduced by only using groups of sensitive taxa within the entire benthic macroinvertebrate community (e.g., a family or order of invertebrates). The choice of a taxon will depend on its sensitivity to the type of stressor.

In the design of the biomonitoring program, it will be necessary to determine the natural spatial and temporal variation in the data so that changes in the data can be attributed to stressor(s). This might require more extensive and frequent sampling at the beginning to determine the natural variation in the distribution and abundance of the taxa. In addition to sampling the biota it will be necessary to sample pertinent physical and chemical characteristics of the aquatic systems depending on the objectives of the biomonitoring.

4.1 Concluding Remarks

The success of the long-term biomonitoring program will be dependent on the choice of an appropriate method and the commitment of resources by EINP over many years. The first three recommendations are important to the choice of an appropriate biomonitoring method. Sufficient resources should be allocated to the analysis and interpretation of the data. This will allow trends to be established and provide an early warning of potential changes in the aquatic systems.

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